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Fish Assemblage Responses to Water Withdrawals and Water Supply Reservoirs in Piedmont Streams

MARY C. FREEMAN*
US Geological Survey
Patuxent Wildlife Research Center
University of Georgia
Athens, Georgia 30602, USA

PAULA A. MARCINEK
Institute of Ecology
University of Georgia
Athens, Georgia 30602, USA

ABSTRACT / Understanding effects of flow alteration on stream biota is essential to developing ecologically sustainable water supply strategies. We evaluated effects of altering flows via surface water withdrawals and instream reservoirs on stream fish assemblages, and compared effects with other hypothesized drivers of species richness and assemblage composition. We sampled fishes during three years in 28 streams used for municipal water supply in the Piedmont region of Georgia, U.S.A. Study sites had permitted average withdrawal rates that ranged from < 0.05 to > 13 times the stream's seven-day, ten-year recurrence low flow (7Q10), and were located directly downstream either from a water supply reservoir or from a withdrawal taken from an unimpounded stream. Ordination analysis of catch data showed a shift in assemblage composition at reservoir sites corresponding to dominance by habitat generalist species. Richness of fluvial specialists averaged about 3 fewer species downstream from reservoirs, and also declined as permitted withdrawal rate increased above about 0.5 to one 7Q10-equivalent of water. Reservoir presence and withdrawal rate, along with drainage area, accounted for 70% of the among-site variance in fluvial specialist richness and were better predictor variables than percent of the catchment in urban land use or average streambed sediment size. Increasing withdrawal rate also increased the odds that a site's Index of Biotic Integrity score fell below a regulatory threshold indicating biological impairment. Estimates of reservoir and withdrawal effects on stream biota could be used in predictive landscape models to support adaptive water supply planning intended to meet societal needs while conserving biological resources.

The ecological effects of meeting the water demands of an expanding human population are of concern worldwide (Postel 2000; Jackson and others 2001). Water withdrawals and diversions used to supply municipalities, industries, and agricultural irrigation have the potential to degrade aquatic habitats to the point that these systems fail to support native biota or to supply other ecosystem services (Moyle and Leidy 1992; Baron and others 2002; Naiman and others 2002). Prominent examples include conflicts between offstream water users and instream flow needs to sustain imperiled species (Collares-Pereira and others 2000; Cooperman and Markle 2003; Ward and Booker 2003), and collapse of fisheries and productivity in flow-deprived ecosystems (Postel 1996, 2000). Even in regions where water historically has been considered an abundant resource, such as eastern North America, rapidly growing populations are placing increasing demands on productive freshwater systems that support unique biodiversity.

The challenges of meeting growing demands for water supply while protecting aquatic ecosystems are exemplified in portions of the southeastern United States. One such area is the southern Piedmont, situated between the Appalachian Mountains and the Coastal Plain, which has experienced some of the highest rates of population growth in the United States in recent decades (Conroy and others 2003; Walters and others 2005). Population growth and urbanization are encroaching on aquatic habitats that support high levels of aquatic biodiversity and endemism, as well as supporting imperiled species (Abell and others 2000; Warren and others 2000). Threats to native biodiversity caused by altered runoff and pollution from urbanizing areas are likely to be compounded by water supply development, largely dependent on surface water in the Piedmont, unless specific management actions are taken to safeguard vulnerable streams. Regulators in the region are attempting to define instream flow

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*Author to whom correspondence should be addressed; *email:* mary_freeman@usgs.gov

needs to protect flowing-water ecosystems, while accommodating societal needs for water.

Management of surface waters tapped for water supply has focused on protecting minimum flow levels, although ecologists have stressed the importance of flows across the range of the natural hydrograph for maintaining structure and function of aquatic ecosystems (Poff and others 1997; Richter and others 1997; Silk and others 2000; Bunn and Arthington 2002). Recent efforts to improve river management have involved developing ecological flow requirements intended to sustain floodplain and in-channel habitats, and recognize flow seasonality and inter-annual variation as drivers of biological communities (Postel and Richter 2003). The holistic approach to defining needs for lotic ecosystems has also shifted the management question from "how much flow must be provided to meet ecosystem needs?" to that of "how much can flow regimes be altered without incurring undesirable ecosystem change?" (Silk and others 2000; Bunn and Arthington 2002). To develop ecologically sustainable water supply policies, regulators will need clear information linking withdrawal levels to effects on aquatic ecosystems.

Previous studies of flow regulation effects on fish assemblages have indicated greater detriment to fluvial specialists, i.e., species that require flowing-water habitats for at least a portion of their life-cycle (Kinsolving and Bain 1993; Travnichek and others 1995) or rheophilic species (Copp 1990), in comparison with habitat generalist species, which are able to maintain populations in lotic and lentic systems. A recent study of fishes in a flow-depleted river in the northeastern United States similarly has revealed a shift in species composition toward habitat generalists and a loss of fluvial specialists (Armstrong and others 2001). Quantifying the responses of differing faunal groups to flow alteration may provide important information to resource managers attempting to balance water use with conserving biota. Additionally, regulatory agencies are often interested in the status of biological communities relative to reference or unimpaired conditions (Barbour and others 1999), in which case effects of water supply development on an assessment score such as the Index of Biotic Integrity (IBI) would be useful.

Our purpose was to improve understanding of the biological effects of water withdrawal by quantifying variation in fish assemblages across streams that are differentially used for municipal water supply. Streams used for water supply vary with respect to permitted withdrawal rate relative to the size of the stream. Withdrawals also vary as to whether they are made directly from an unimpounded stream or from an instream reservoir. Reservoirs, by trapping and storing water during periods of higher runoff, potentially alter downstream flows over a broader range of the flow regime than direct withdrawals. Thus, we investigated the effects of increasing the relative withdrawal rate and the use of water-supply reservoirs on stream fish assemblages. We specifically examined effects on richness of fluvial specialist (FS) and habitat generalist (HG) fishes, and compared the influences of withdrawal rate and reservoir presence with effects of three site-level variables chosen to represent influences of natural (drainage area, bed sediment size) and anthropogenic (amount of urban land use) factors on fish assemblages. We also asked whether sample-specific instream habitat conditions improved site-level models for predicting species richness. To analyze assemblages relative to reference conditions, we used multivariate ordination to analyze assemblage similarity between our samples and samples taken in Piedmont reference streams, and also evaluated IBI scores for our samples in relation to a regulatory threshold used to indicate biological impairment. We used results to estimate quantitative effects of increasing water allocations and using instream reservoirs on stream fish assemblages, and considered how these estimates could be used to incorporate biodiversity conservation in water supply planning.

Methods

Study Site Selection

To hold other landscape influences more constant, we restricted the study to one physiographic area, the lower portion of the Piedmont physiographic region of Georgia (Figure 1). We used a GIS database of 53 permitted water withdrawals, obtained from the Georgia Department of Natural Resources, Environmental Protection Division (GDNR, EPD), to identify potential study sites within 6 river systems (Savannah, Ogeechee, Oconee, Ocmulgee, Flint, and Chattahoochee). We selected 27 study sites in 2000 that represented all of the wadeable streams with apparently active withdrawals and drainage areas exceeding about 12 km^2 (corresponding to the smallest reference site sampled in 2000 by the GDNR Stream Survey Team). In addition to sites located in non-wadeable streams, sites that were dry (2), affected by ongoing construction or pump-storage operations (2), or that were not accessible (2) were not included in the study. Of these 27 sites, 13 were situated downstream from direct water withdrawals ("intakes") and 14 were located downstream from water supply reservoirs ("reservoirs"; Appendix 1). We

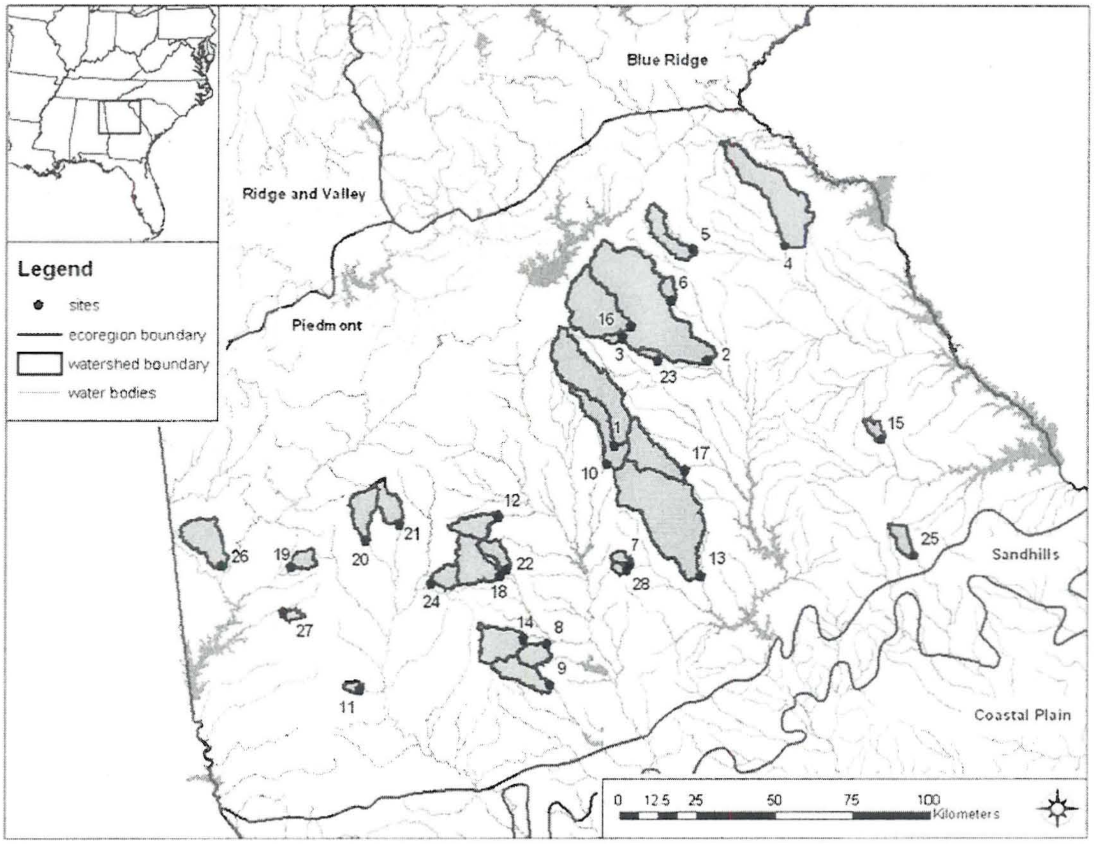

Figure 1. Watershed boundaries for 28 municipal water supply withdrawals in the Piedmont ecoregion, and adjacent ecoregion boundaries. Numbers correspond to sites listed in Appendix 1.

repeated sampling in two subsequent years, 2001 and 2003, to assess effects over a range of instream habitat conditions. For the 2001 field season, we added one intake site that was dry during 2000 and eliminated one intake site that became non-wadeable. During 2003, which had substantially higher rainfall than in 2000 and 2001, we were able to sample 20 (12 reservoir sites and eight intakes) of the 28 sites sampled in the previous years; the remaining eight sites were not sampled because they became non-wadeable under the higher flow conditions.

The GDNR sampled streams chosen as Piedmont stream reference sites (i.e., for bioassessment purposes) in 2000 and 2001 and made those data available for comparison to our study sites. Reference sites were chosen by GDNR on the basis of appearing relatively unimpaired and supporting relatively intact fish assemblages, and were located within the lower Piedmont in five of the six river systems containing study sites. None of the sites were directly downstream from withdrawals or reservoirs. Seven Piedmont reference sites were sampled in 2000, ranging from 12.3 to 690 km^2 in drainage area. These seven and two additional sites (9.7 and 20 km^2) were sampled in 2001. The GDNR did not sample Piedmont reference sites in 2003.

Sampling Procedures

We sampled withdrawal sites between June and September in each study year; reference sites were sampled during September and October in 2000 and 2001. The June through October timeframe represented a period of relative assemblage stability (Matthews 1990; Peterson and Rabeni 1995) occurring after the spring period of spawning migrations by some species, and was within the sampling period (April through October) used by GDNR to assess integrity of stream fish assemblages. We sampled fishes following protocols developed by GDNR (GDNR 2000). Reach length approximated 35 times mean wetted-channel width (estimated from width measurements at five randomly chosen locations for each 100 m length included) to a maximum of 500 m. We sampled fishes from the downstream to upstream boundaries of the sample reach using one or two backpack electrofishers, or a barge-mounted boat electrofisher (if mean stream width exceeded about 8 m and the stream was deep throughout), dip nets, and seines. Captured fishes were transferred to buckets and coolers; fish to be measured and released were kept in aerated, frequently exchanged water. Fish that were preserved were first anesthetized with tricaine methanesulfonate (MS-222) and subsequently transferred to 10% formalin. All fish were identified to species and measured (total length) either in the field or laboratory. We also recorded incidences of individuals with evident disease, eroded fins, lesions, and tumors, and with "black spot" (trematode cysts).

We measured stream discharge, water temperature, and turbidity on each sampling date. Turbidity was measured in nephelometric turbidity units (NTU) with a Hach® Model 2100P turbidity meter. In 2000 and 2001, we also measured dissolved oxygen (DO) concentration at the upstream end of each sample reach with a Hydrolab® multiprobe. We estimated stream discharge near the upstream end of the site using a Marsh-McBirney Flo-Mate® electromagnetic current velocity meter and top-setting wading rod. Discharge was based on depth and velocity measured every 0.5 m along a tape measure stretched from bank to bank, or at intervals sufficient to give at least 20 measurements across the stream.

Habitat data were collected during low flow conditions, usually immediately following fish collections. The length of each pool, riffle, and run in the sample reach was measured and recorded in sequence by type. We randomly selected three to five locations for cross-section measurements from the total length of each habitat type. At each cross-section, we recorded depth, velocity, and dominant bed sediment size [recorded in *phi* intervals (Gordon and others 1992)] at channel edges and at three equally spaced positions across the channel (0.25, 0.5, and 0.75 times wetted width). Depth, velocity, and bed sediment averages for each site were computed as means of pool, run, and riffle measurements, weighted by the proportion of the site in each habitat type. In 2003, dominant bed sediment size was recorded longitudinally along the mid-channel at intervals equal to 0.5 times average wetted-channel width to give a more complete profile of sediment variation.

Analyses

Site Characteristics. We summarized average stream flow, depth, velocity, and water quality measurements to compare flow and habitat conditions among years and between intake and reservoir sites. Land use in the catchment upstream from each site was estimated using a statewide land cover map based on 1998 *Landsat* Thematic Mapper imagery, with 30-m resolution (produced by the Natural Resources Spatial Analysis Laboratory, University of Georgia, in 2001). We examined three land use categories: urban (low- and high-intensity), forest (including deciduous, evergreen, and mixed forest), and agriculture (pasture and row crop). We estimated mean bed sediment size at sites by averaging the mean values computed for each year the site was sampled.

To facilitate among-site comparison of potential withdrawal rate relative to stream size, we computed a "withdrawal index" (WI) for each site as the maximum permitted monthly average withdrawal rate (in million gallons per day, mgd) divided by the estimated seven-day, ten-year recurrence low flow (7Q10) at the withdrawal site (also expressed in mgd). The WI thus represented the fraction or multiple of the 7Q10 flow permitted for withdrawal on a monthly average basis. The 7Q10 flow is commonly used by regulatory agencies to set wastewater discharge criteria, and has also been used to set minimum flow requirements. We used 7Q10 flows to standardize permitted withdrawal rates across sites because 7Q10 estimates could be obtained from EPD files or estimated from low-flow profile data (Carter and others 1986, 1988a,b). Other flow statistics that could also be used to standardize withdrawal rates, such as average annual flow, were not available because 27 of the 28 sites lacked stream gages. This also precluded quantifying actual hydrologic patterns or flow alteration. To examine actual water withdrawal in relation to WI, we used the maximum monthly withdrawal rate reported for 25 sites during the 12-month period prior to our fish samples in 2000 and 2001, divided by 7Q10 for the site. Monthly withdrawal data were provided by EPD or by permit holders. Because we observed strong relations between WI and water use (results reported below), and because water use varied at many sites across months and water use data were not available for all sites and months, we used WI as our measure of potential withdrawal effects in all analyses.

Fish Assemblage Patterns. We used the catch data across sites and years in an ordination analysis to examine the relative similarity of fish assemblage data in our samples to those from the GDNR Piedmont reference sites. We employed nonmetric multidimensional scaling (NMDS), using Bray-Curtis dissimilarity, to ordinate our study-site samples (n = 74) and Piedmont reference site samples (n = 16). NMDS provided graphic representation of the relative similarity among samples based on taxa abundances (Field and others

1982; Clarke 1993; McCune and Grace 2002). For each sample, we applied a fourth-root transformation to abundance data in order to dampen the influence of common taxa (Clarke 1993), retaining for analyses all taxa that occurred in at least six samples overall and that had ranges encompassing all sample locations. To meet the latter criterion, we combined abundances of some congeneric species with ranges restricted to a subset of the sampled river systems (Appendix 2). We used a total of 34 taxonomic entities (24 species and 10 multispecies genera) in NMDS analysis. Analyses were accomplished with PC-ORD (McCune and Mefford 1999), using a step-down procedure (from 6- to 1-dimensions) to find the most appropriate solution, with 200 iterations, 10 runs with the real data, 20 runs with randomized data, and stability criterion set to 0.0001 (McCune and Grace 2002). Because of reviewer concerns that the PC-ORD solution might be far from the minimum stress solution, we also conducted NMDS ordination using the function isoMDS, package vegan, in the R programming environment (Oksanen and others 2005), using Bray-Curtis dissimilarity and the same number of dimensions (3) as in the final solution from PC-ORD. We used product-moment correlation coefficients between taxa abundances and ordination axes, and graphical representation, to examine patterns of assemblage differences among sites.

Water Withdrawal and Reservoir Effects on Species Richness. We used the limiting form of the jackknife estimator for model M_h (Burnham and Overton 1979; Williams and others 2002) to estimate richness of FS and HG species in each sample, given the observed numbers of species and numbers of species in each sample represented by 1, 2, 3, 4, or 5 individuals. We used the program SPECRICH, available at http://www.mbr-pwrc.usgs.gov/software.html. The purpose of using the jackknife estimator for richness rather than actual sample counts of species was to reduce bias resulting from incomplete species detection and among-species differences in detectability.

We used an information-theoretic approach (Burnham and Anderson 2002) to evaluate the relative effects of WI and upstream presence of a reservoir on fish species richness. Our approach was to construct a set of alternative linear regression models that, first, would allow us to compare the effects of WI and reservoir presence with each other and with other site-level variables hypothesized to influence species richness. Secondly, we wished to evaluate effects of among-year differences in instream habitat on species richness. Our data set comprised 72 observations (one to three observations at 27 sites; see Results). To avoid model over-fitting, we restricted models to a maximum of 7 parameters (i.e., to keep the ratio of observations to parameters to about 10 to 1; Burnham and Anderson 2002). All regression models included a term for random variation among sites to account for unmeasured site-specific influences on the repeated observations (Snijders and Bosker 1999) in addition to an intercept and within-site residual error term. Thus, we included a maximum of four explanatory variables in regression models to limit the total number of model parameters to seven.

We considered the effects of three site-level variables in addition to WI and reservoir presence on species richness for FS and HG fishes. First, we included drainage area in all models because fish species richness generally increases as a function of stream size (Matthews 1998). Given that our sites spanned two orders of magnitude in drainage area, we did not believe that any credible explanatory model could ignore drainage area as a predictor variable. We also hypothesized that the average size of the stream bed sediments would influence species richness of FS and HG fishes, based on observations in upper Piedmont streams of a shift in fish assemblage structure from dominance by fluvial specialists in steeper, rockier streams to habitat generalists in lower gradient streams with finer bed sediments (Walters and others 2003b). Finally, we hypothesized that the level of urbanization upstream from the study sites could depress species richness, at least of FS species, as observed in other Piedmont streams (Weaver and Garman 1994; Walters and others 2003a; Roy and others 2005). Therefore, we evaluated 13 models with drainage area and combinations of WI (ln transformed), reservoir presence, mean *phi*, and percent urban land use upstream from the site (arcsine transformed). We also evaluated whether adding terms indicating basin identity, presence or absence of a minimum flow requirement, and an interaction between WI and reservoir presence improved the best-fit site-level model predicting species richness. Our purpose was to construct a small set of preselected candidate models (Burnham and Anderson 2002) intended specifically to compare effects of WI and reservoir presence with variables representing natural and land use influences, while also testing for the potential influences of differences attributable to basins, minimum flow requirements, and the possibility of a reservoir-withdrawal level interaction. Basin identity was coded as Apalachicola (i.e., Chattahoochee and Flint river systems), Altamaha/Ogeechee (i.e., Ocmulgee, Oconee, and Ogeechee river systems), or Savannah following Warren and others (2000).

The question of protective minimum flow levels remains unanswered, except to note that there is no evidence that providing for a minimum flow of 7Q10 protects stream fish assemblages, either from our data or more generally (Stalnaker and others 1995). Higher minimum flow provisions may mitigate some effects of withdrawals and reservoirs, but only if periodic low-flow depletion is the primary pathway by which hydrologic alteration influences stream biota. If biotic integrity is diminished by flow reduction during periods of normally higher base flows, then requiring a protected minimum flow level will be insufficient to protect stream ecosystem integrity (Poff and others 1997; Richter and others 1997).

The results of this study support two hypotheses that could be applied, tested, and refined through the process of developing water supply in the rapidly growing regions of the eastern United States. Our results indicate that (1) increasing permitted water withdrawal levels is likely to result in local loss of stream fish species, specifically fluvial-dependent species, and (2) construction of instream water supply reservoirs is similarly likely to result in reduced richness of fluvial-dependent species. Based on our data, streams in the lower Piedmont may begin to experience species losses if permitted withdrawal exceeds about 0.5 to one 7Q10-equivalent of water. Additional research to broaden the geographic scope and size of the data set could improve our ability to predict effects of water withdrawal and use of reservoirs on stream fishes. However, given the ecological complexity of stream systems, i.e., structural uncertainty (Williams and others 1996), and the difficulties in precisely quantifying the richness and abundance of many stream species (i.e., partial observability), there likely always will be considerable uncertainty when predicting the effects of any given withdrawal or reservoir on stream biota.

Taking an adaptive management approach (Walters 1986) to future water supply development could allow communities to meet their water needs while working with managers and regulators to conserve the biological diversity native to a region's streams. Conroy and others (2003) argue that, given uncertainty and the high ecological stakes of current development trajectories in areas such as the southern Piedmont, management should be based on adaptive decision making utilizing predictive models that relate policy decisions to integrity of stream ecosystems at varying scales. Estimates such as generated in this study could be useful in beginning this process. For example, in the Piedmont region of Georgia, managers could use our results to hypothesize that withdrawals exceeding a given rate are likely to result in species losses, and that supplying water by way of multiple, dispersed withdrawals capped below that level will have fewer effects than concentrating supply at large withdrawals or instream reservoirs. Decision makers could identify streams that appear "over-allocated" with respect to supporting native fishes, and test and refine this hypothesis, as well as identifying areas within basins where further allocation is likely to lead to faunal decline. Decision makers could also evaluate alternative supply scenarios with respect to predicted biological effects given this hypothesis, preferably in the context of a predictive model incorporating other influences such as changing land use. Decisions regarding individual projects will be influenced by multiple factors, including the presence of rare or imperiled stream biota (e.g., species protected under the Endangered Species Act or Georgia's Endangered Wildlife Act) and economic considerations, but whatever decisions are made, one could predict effects on biological integrity in the affected stream systems. Importantly, monitoring stream biota before and after implementation of new withdrawals could then test those predictions, with the results used to improve our understanding of relations between withdrawals, water supply reservoirs, land use change, and stream biota. Applied at a regional scale, water supply development could be planned to avoid excessive depletion and fragmentation in stream systems critical for supporting unique faunal assemblages. This approach would differ from that currently taken by shifting the emphasis from minimum flow policy and provisions, to the adaptive development of water supply strategies that conserve biological resources.

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